Attachment 3

SYSTEMATIC
MANAGEMENT
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29 October 1991

Fraser R. Lockhart, Director
Environmental Restoration Division
Department of Energy
Rocky Flats Office
Golden, CO 80402-0928

Re: Review of Baseline Risk Assessment and Environmental Evaluation Sections of:

(1) October 1991, Preliminary Draft phase I RFI/RI Work Plan for OU10 (other outside closures) at Rocky Flats Plant (3 volumes)

(2) June 1991, Final Phase II RFI/RI Work Plan (Bedrock) , 903 Pad, Mound and East Trenches Operable Unit No. 2) Rocky Flats Plant

Dear Fraser:

I received the subject documents within your October 16th package to me upon my return from RFP, on October 28. Since the items listed above have a deadline of today I am providing my comments to you following only a day's review of the two. I have also not been able to reach HAZWRAP staff to discuss their comments. Evidently, they are in meetings today; the phones have not been answered at either Randy Harris or Tom Brennan's office.

Both Documents:

1. The methodology to be used for risk assessment (actually human risk assessment) and for environmental evaluation (actually population and community evaluation) is much less developed than the balance of the Work Plans. There is an implied or explicit promise [see page 8-13 of the OU 10 work plan] that the exposure scenarios will be developed, assumptions stated, and data use specified at some future point and submitted to EPA for review and approval. I have the following concerns:

   A work plan is being submitted which will support the development of a health risk assessment and environmental evaluation under baseline [that is pre-remedial actions] conditions but we do not define in other than the most generic of ways the following:

      data needs including parameters for measurement, frequency
      data quality objectives
      the level(s) of resolution analyses will be performed
      specific methods to be used for assessment
      levels of risk [impacts] to be resolved/predicted

"Controlling The Future"

What is the relationship between those data specified in field sampling/analysis of contaminants with health risk assessment and environmental evaluation?

What is the relationship between the work plans and the ongoing efforts in risk assessment/modeling?

2. Based on the current knowledge of the ecosystems at RFP [at least representative of RFP aquatic and terrestrial systems based on the myriad of studies evaluating ecosystem, community and population dynamics by CSU and others], it seems we could in the work plan be able to clearly establish the following:

What are the most probable [e.g. > 75%] target species exposure scenarios?

What media of contaminant accumulation are likely? What are the likely biomagnification routes? What nutrient cycling processes are likely to be affected?

What are the expected complexity, stability, and diversity parameter values of representative ecosystems? How do these differ from estimates made under current conditions?

3. Based on current epidemiological data for the area [both occupational exposures for DOE plants and RFP specifically and the public in the communities of Boulder, Golden, Denver]:

What are the most probable [e.g. > 95%] human exposure scenarios?

What are the most probable off-site exposure scenarios?

What are the ambient levels of illnesses in the geographic region?

What are the ambient incidence of death which could be caused by principal contaminants of RFP in the geographic region?

Recommendation

It would seem to me that it would be very appropriate to develop a set of SOPs for risk assessment and for environmental evaluation for all OUs on at RFP. The SOPs could then address many of the issues which are discussed below and the OU work plans could cite these and specify their application in the specific OU study.

I believe it is critical that we very rapidly define methodologies to be used and levels of detail (order of magnitude, qualitative, quantitative) which can be used as the basis for specific work plans. Within each of these levels, it is possible to identify a variety of assessment tools, including models, that may be used, depending on the data requirements, purpose of the assessment, etc.

If such an approach were taken, the risk assessments and environmental evaluations which will need to be completed for proposed remedial actions during construction, operations, and following closure or during long-term monitoring for both NEPA and CERCLA purposes will be much easier to complete and more consistent with the baselines prepared during this phase.

Referenced Document 1: Section 8, Baseline Risk Assessment

1. The material presented in Volume 1, pages 5-4 through 5-6 is totally duplicative of that presented in Volume 2, pages 81, 8-2 and 8-5. It seems appropriate that this material be in Volume I and that a more detailed discussion focusing on the implementation of the general methods sanctioned by EPA be placed in Volume II.

2. The risk assessment overview (p 8-1 Volume II) states that the baseline risk assessment will serve as justification for doing remedial actions. It would seem more appropriate to discuss the role of the baseline human health risk assessment presented in Section 8 and its role in determining need for remedial actions when coupled with (1) ARARs, (2) effluent and emission limits, and (3) environmental (ecological) risk assessment.

   The last sentence of paragraph one cites the EPA guidance for potential human health impacts "as part of" this baseline risk assessment — we fail to clearly define any other parts and it is not apparent to the reader that there are any.

3. Section 8.1, page 8-1 bullet #3 — why are we defining environmental receptors here since the rest of Section 8 deals with human exposure/risk? Is this meant to be referring to exposure to media such as soil resuspended dust having inhalation risk, water as ingestion risk or to food stuffs e.g plants/animals who have had significant uptake and retention of contaminants?

If so, please clarify. If this refers to ecological risks I'd recommend we delay its discussion to Section 9 where it is treated more fully.

4. Section 8.1, page 8-2 first full paragraph -- redundant with paragraph one of the section. We could state the objectives specifically for the baseline human health risk assessment in this space.

5. Section 8.2, page 8-2 first sentence -- the objectives stated are extremely vague. It will be difficult to gather "all OU10 data relevant." The objectives

should specifically identify those data collection and data evaluation tasks required to satisfy the risk assessment to be produced.

6. Section 8.2, page 8-1 paragraph -- It is stated that data were collected, more is being collected and all will meet QA criteria. However, it is not obvious what is now known that provides the basis for additional data collection as well as its justification to cause expenditures. It would be helpful before beginning this section to assess what the data available contribute to the risk assessment, what is missing, what has to be done to fill the data gaps and how essential these data are for risk assessment purposes.

7. Table 8-1: Reference to EPA 600-3/89/013 seems out of place in the human health assessment section as does EPA/540/1-89/001A. While there is an interface with the environmental evaluation, it seems these citations are directly applicable to Section 9. Conversely, why are there no citations to general human health risk assessment methodologies -- reliance on the EPA guidance is problematic since they are dated and the agency assumes that the risk assessment staff routinely are aware of new developments in the field and the wider knowledge basis than what provides their "benchmark" guidance.

8. Section 8.2 pages 8-6 and 8-7 -- many of the items listed could have and possibly should have been done prior to the development of the work plan to quantify parameters needed for risk assessment. Assuming that a section such as that recommended in comment 6 above is created, the information here could be cast as the second tier needed to get from order of magnitude risk evaluation to at least quantitative and for some COC, quantitative risks.

9. Section 8.3 -- it would seem appropriate to state what is known, what hypotheses have been developed and the process that will be followed

specifically to test these with respect to exposure; the current section reads as a generic treatment of the process with no information furnished as to the work being planned specifically for OU10.

10. Section 8.4 -- it seems still too generic to be a work plan for a specific OU. The use of human toxicology data and testing data from in vitro and in vivo animal models should be distinguished and the reliance of the risk assessment on the two stated.

11. Section 8.5 — How is synergistic risk assessed? Do you envision one characterization per COC? If so, how do you integrate for total risk?

12. Section 8.6 does not define the work plan to be implemented from the standpoint of uncertainty. The following as a minimum should be addressed:

    what sources of uncertainty are recognized

    how do the uncertainties ramify through the assessment process (in other words are they additive, multiplicative, do we have a basis to know)

    there are uncertainty estimates for epidemiological data, for toxicity data, for extrapolation of animal model test data to humans, etc.

<u>Referenced Document 1: Section 9: Environmental Evaluation Work Plan</u>

1. General to Section 9.0: RFP/RFO staff has had many discussions about ecosytems and use of a "systems" approach to conducting environmental evaluations especially in the risk assessment committee. In the first paragraph, it is apparent that an attempt has been made to include this approach in the evaluation methodology. However, the paragraph demonstrates that the concept is not understood and the balance of the section excludes any but biotic impacts with the exception of an analytical approach to evaluation of trophic structure.

    It seems appropriate that one of two directions is taken immediately. Either the approach used is : (1) <u>system</u> [which means that one or more indicators of effects are evaluated at all levels of ecological complexity, including systems, community, population, leaving organism level and biochemical level impacts in the domain of toxicology and reflected only in ramifications into trophic structure changes or as analogs to human health risks] or (2) biotic [which means that analysis is made of toxicological and population impacts of target species which are T&E/protected, key from a trophic perspective or are economically or recreationally important].

The argument for use of a systems approach is that it provides a mechanism to determine stress and recovery of aquatic and terrestrial ecosystems in response to chemical contamination. Since systems parameters represent measurements of the highest order interactions within the system [in this case ecosystem] they are sensitive to synergistic stresses and indicative of biological effects that will be manifested at higher contamination rates or after increasing exposure lengths. Parameters to monitor include those measuring nutrient cycling, system elasticity/resilience, and complexity. Of these nutrient cycling parameters are the best documented, understood and most cost effective to use.

RFO/RFP must make the decision which approach to follow. My concern is that the DOE [actually its predecessor agency AEC] made a large investment in the development of ecosystem theory, especially in ORNL's Environmental Sciences Division. The investment is reflected in much of the ecological literature which used the cycling of radionuclides as the basis for construction of much of the theory, the analytical and mathematical models, and the field verifications of them under chemical and other stresses (e.g. clearcutting, tree girdling). Within the DOE programs it would be appropriate to (1) use all the theory and data collection/analysis techniques developed applicable to risk assessment and environmental evaluation and (2) not present information which erroneously applies the developed work.

If a systems approach is used [and I would strongly recommend doing so] then I strongly recommend the basis for its use be founded in work done by O'Neill (ORNL), Shugart (now at VPI), Waide and Webster, and a cast of others primarily based in Oak Ridge, U GA and CSU. I have enclosed a copy of a 1983 paper I published in Environmental Monitoring and Assessment, primarily for the bibliography.

2. Section 9.1; Paragraph 1: The phrase..."quantifying the ecological effects to the biotic environment..." does not make sense to the reader. The proposal to address ..."all ecosystem components..." is an invitation to failure as well as excessive expenditures for our purpose. I suspect that the intent was to

address indicators of the structural complexity of the systems but even that represents a large body of work that is difficult to justify to support for risk assessment and environmental evaluation purposes--- for one thing it is difficult to show what you'd do with all the data if they were collected.

If a systems approach is desired, I would recommend that the following parameters be used and the relationship to risk assessment/environmental evaluation techniques be carefully defined before field work is begun:

ecosystem	nutrient cycling as indicated by export rates of essential elements along a gradient from highly contaminated to background COC levels within a system (a subwatershed not a vegetation type or habitat)
community	trophic analyses, as briefly indicated in Section 9.2; parameters which can be used include analysis of redundancy in each trophic level [so that food availability may be less affected if one food species for higher level populations is affected], identification of "kingpin" species [those which if adversely affected cause significant changes in higher order populations be out migration of those species, declines due to lack of food or changes in food species taken]
	diversity, especially of plant species
population	dynamics [e.g reproduction rate, numbers or biomass]
	toxicological response, including biouptake and bioaccumulation

I would highly recommend no attempt be made to evaluate microbial communities except as they are reflected in nutrient cycling measures.

3. Section 9.1, page 9-3 second paragraph — this will be the outcome only if a systems approach is followed. I concur it the desired outcome and it provides the linkage to human risk assessment as well.

4. The balance of the work plan seems adequate to evaluate biotic effects within the impacted systems. The use of trophic analysis is not defined at the level allowing review. It seems the use as presented is only to identify target species for tissue analysis and ecotoxicological evaluations.

Referenced Document 2: Section 5.6: Baseline Risk Assessment

1. The level of detail is inconsistent with that prepared for OU10.

2. The organization of health risk and environmental evauation as components of a single baseline risk assessment is preferable, in the opinion of the reader to their use as separate analyses as was done in OU10.

3. Since the subject is bedrock would it be preferable to describe how the data being collected for this phase is (1) integrated into the OU database and (2) contributes to filling the data needs for the OU baseline risk assessment? It would seem reasonable to prepare no more than one integrated baseline risk assessment including public health risk and environmental risk for the OU and to include in this work plan only the relevant data needs, parameters to be measured, etc.

4. The environmental evaluation (Section 5.6.2, page 5-9 & 10) is clearer than that presented in OU 10 only because it makes no attempt to incorporate the systems approach to the analysis. Comments from the OU10 work plan above (all of comment 1 and the last of comment 2 describing parameters for measurement) apply to this discussion as well.

5. If the general comment recommending the development of SOPs for the risk assessment (including health and environmental) were adopted, the work plan discussion would be the brevity of the current discussion [which is helpful] and be specific to the work to be completed to support the assessments by its implementation.

If I can be assistance in revising these risk assessment sections, please let the staff know of my willingness and availability. If a systems approach is to be adopted, I would willingly assist development of the SOPs, etc, since ecosystems analysis is my specialty and emains a keen interest, especially as it applies to remediation and D&D projects.

Thanks for the opportunity to review and comment on these work plans.

Sincerely,

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AN ARGUMENT FOR ECOSYSTEM LEVEL MONITORING*

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(Received 23 February, 1983)

Abstract. One objective of environmental monitoring programs is documentation of qualitative and quantitative environmental changes in response to external stresses, including chemical contamination. Chemical contaminant, biological, and ecological measurements have been used as environmental monitors. Contaminant monitoring allows estimation of exposures, biological and ecological monitoring allow estimation of uptake and effects.

Measurements of ecosystem homeostasis such as nutrient cycling processes have been shown to be good ecosystem level monitors. The rate of dissolved nutrient loss from ecosystems has been conclusively shown to increase as a function of chemical contamination until a new equilibrium is reached, the pollutant input has become negligible, or until nutrient pools have been depleted. Consequently, nutrient pools in environmental strata and in biota are altered and eventually depleted by chemical stress.

The use of nutrient cycling to determine sensitive responses to and long-term changes for chemical contamination is an essential monitoring strategy for environmental management and compliance purposes. Measurements of export (rapid response) and pools (long-term consequences) are within current technology, are cost-effective, and allow rapid implementation of remedial measures or environmental controls.

1. Introduction

There is increasing public awareness of the direct and indirect detrimental effects of contaminants on terrestrial ecosystems. Even the subtle effects of contaminants may be manifested after many years of exposure (e.g., acidic precipitation effects on productivity or the spread of toxic chemicals from buried waste). The public needs reassurance that continued industrial development, environmental decontamination and control technologies applied for containment of contaminants do not necessarily impart irreversible degradation of environmental quality and degeneration of human health. In fact, the public demands evidence that these activities do not cause significant human exposure or increased environmental degradation.

Documenting changes in environmental concentrations of contaminants and resultant biological/ecological effects have been largely legislatively mandated. The U.S. National Environmental Policy Act (NEPA, P.L. 91–190) requires that knowledge of existing conditions be established prior to major activity and that changes during and following these activities be established. Conditions set forth in the Clean Air Act (P.L. 88–206) and the Water Pollution Control Act (U.S. Code 1970, Title 33, Section 1171) require repeated measurement of pollutant loading and distribution. Specific license or permit-requiring legislation (e.g., National Pollutant Discharge Elimination System in the Water Pollution Control Act), the Resource Conservation and Recovery Act (RCRA,

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P L 94-580), Federal Insecticide, Fungicide, and Rodenticide Act (FIFRA, P L 92-516), and the Toxic Substances Control Act (TSCA, P L 469) demand or imply monitoring of inputs, distributions, and/or biological/ecological responses

Additional motivation for developing a viable monitoring strategy has come from citizens' action groups Experience of the last few years has shown that compliance with existing regulations does *not* necessarily assure public health and safety or environmental quality Meeting existing regulations does not necessarily guarantee that industries will not be held accountable for present environmental release at some future time For example, the contributors to toxic waste disposal facilities are being financially responsible for the remedial measures required to contain these wastes away from environmental media

The U S Environmental Protection Agency (EPA) is attempting to implement the host of congressional mandates by testing chemicals prior to release and monitoring contaminants in the field Increasing emphasis is being placed on monitoring ecological consequences in the field This approach is due to the lack of adequate laboratory testing protocols (under P L 469) and the lack of knowledge of potential pollutant synergistic interactions which may occur in the environment

This paper has as its objective to present the theoretical and experimental basis for an ecosystem level monitor and to argue for its use as the primary tool to continually gage environmental quality

2. Theoretical Basis for Ecosystem Monitoring

An ecosystem consists of coupled synthesis and degradation processes which allow a persistent recognizable biological assemblage to persist through time (Weiss, 1971, Blackburn, 1973, Waide and Webster, 1976) Figure 1 shows an extremely simplified diagram of ecosystem function There are two reservoirs of raw materials, atmospheric and geologic Together with precipitation and anthropogenic materials these provide inputs to the assemblages of biogeochemical processes called ecosystems (Odum, 1971). That assemblage can be viewed simplistically as a synthesis system integrating carbon, nutrients, and selected anthropogenic inputs into a wide variety of compounds within biological and abiotic components (e g , soil, vegetation)

Taken one step further, Figure 2 shows a conceptual diagram of the ecosystem The larger circles represent ecosystem components such as litter and vegetation (i e , pools of organic and inorganic compounds) Each component interacts with others via processes like consumption, nitrification, or hydrolysis The processes are integrative and synergistic minimizing exports of essential elements via gaseous, solute or erosive losses (Borman and Likens 1979)

While Weiss (1971), Odum (1971), Bormann and Likens (1979), O'Neill *et al* (1977), and Schindler *et al* (1980) have repeatedly emphasized that the ecosystem should be a primary focus of experimental studies, the state-of-the-art for pollution studies has remained at the population and community level of organization The lack of scientific concentration on the integrated, biochemical responses of an ecosystem to pollutant

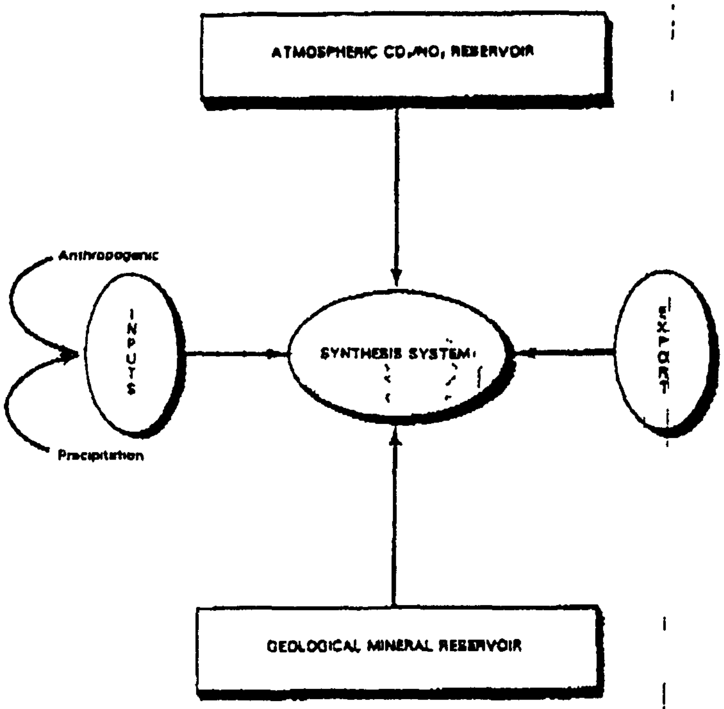

Fig. 1 A highly simplified diagram of an ecosystem.

stress is the major barrier to development of a viable terrestrial monitoring program applicable across the United States

Changes in population and community dynamics can occur with little effect on ecosystem level processes For example, over a time period, a dominant producer population might be replaced, but the primary production process might remain relatively constant (O'Neill and Giddings, 1979) Studies of species or community effects of a pollutant made in the laboratory or in the field cannot lead to inferences of ecosystem responses to that pollutant because (1) the components of an ecosystem behave differently when isolated from it (Rosen, 1972, Morowitz, 1966, McNaughton, 1977, O'Neill and Waide, in press) and/or (2) compensatory mechanisms may attenuate responses of components studies with a system

The crux of ecological theory applicable to monitoring is to approach it as it exists, as a biogeochemical unit (Hutchinson, 1948, Lane *et al*, 1975, Schindler *et al*, 1980). Measurements made, then, should reflect the autotrophic-heterotrophic couplings (Odum, 1971, Ausmus *et al*, 1977) rather than sets of species of organisms. For example, nutrient cycling and retention in terrestrial ecosystems result from the functional synchrony of autotrophs, heterotrophs, and soil biochemical matrices (Harris *et al*, 1974, O'Neill *et al*, 1975, Overton, 1975, Reichle *et al*, 1975) By focusing on such

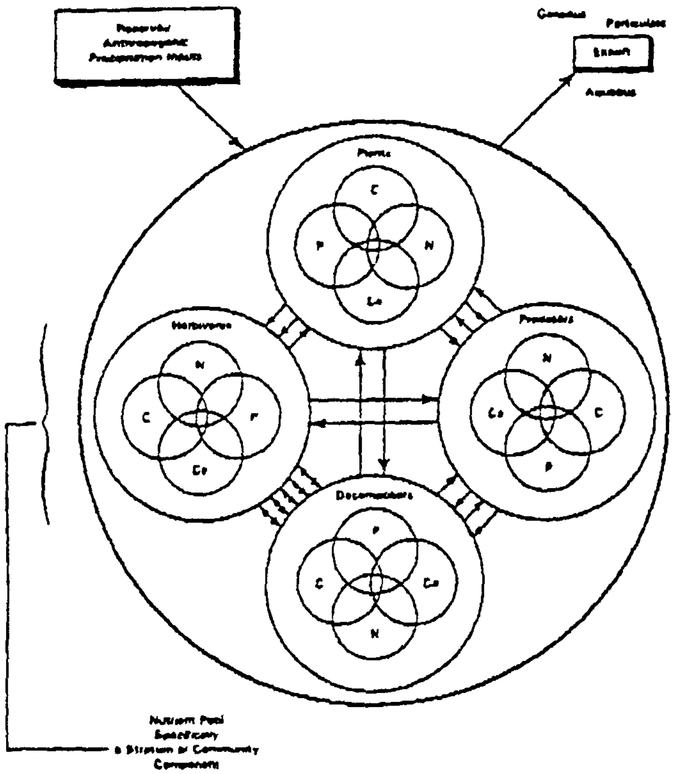

Fig 2 Conceptualization of a soil showing nutrient pools and exports.

interactive processes, it is possible to identify, evaluate and employ monitoring points reflecting short- and long-term changes in the total terrestrial ecosystem (Patten, 1975)

Nutrient export rates (O'Neill *et al*, 1977) are obvious nutrient cycling monitoring points in terrestrial ecosystems because these exports reflect the interactions of biological and chemical process However, they may not reveal the mechanism(s) causing the change That is, while nutrient flux may be the most sensitive indicator of stress because it is the highest order interaction within the system, the specific causes manifesting biochemical and/or biological changes will not be obvious from these measurements alone. Application of control theory in forest ecosystems studies supports such an approach, suggesting nutrients in soil solution as an optimal monitor of the forest status (Shugart *et al*, 1976) An optimal monitor in the desert might be gaseous nitrogen cycling using measurements of fixation and/or denitrification (Ausmus and Dodson, 1978)

While nutrient export rates offer potentially a very sensitive index of ecosystem status, they respond to all stresses, including natural environmental short-lived phenomena (e g, freeze-thaw events) (McGill, 1980) Chronic stress causing a degradation in the

3 Experimental Basis for Ecosystem Monitoring

In terrestrial ecosystems, nutrient cycling processes are dominantly present in the rooting zone, controlling the rate of nutrient export Except in tropical systems where litter interactions dominate, soil is the matrix of these interactions (Alexander, 1961, Gardner and Ashby 1970, McGill, 1980) The pools of nutrients and their chemical forms in all terrestrial ecosystem components then are the results of these interactions

TABLE I

Summaries of several studies showing the effects of both chemical and nonchemical stresses on terrestrial nutrient export rates

Ecosystem	Stress applied	Nutrient export response	References
Forest	Clearcutting (field)	Increased K Ca, Al, Mg Na NO_3–N loss	Bormann and Likens, 1979
	Tree girdling (field)	Increased NO_3–N and Ca loss, increased CO_2 loss	Edwards and Ross-Todd, 1979
	Soil trenching (field)	Increased NO_3–N loss	Vitousek 1977 Vitousek *et al* 1979, Vitousek and Melillo, 1979
	Lead smelter emissions (microcosm)	Increased Ca, Mg NO_3–N and CO_2 loss	Jackson *et al*, 1977, Ausmus *et al*, 1978
Forest Soil	Hexachlorobenzene (microcosm)	Increased CO_2 loss	Ausmus *et al*, 1979
Grassland	Arsenic (field and microcosm)	Increased PO_4–P, NO_3–N, NH_4–N and DOC loss in microcosms	Jackson *et al*, 1979
Forest and grassland	Acid rain (field)	Increased cation leaching	Abrahamsen *et al*, 1977
Desert	Cadmium (microcosm)	Increased CO_2 loss	Ausmus and Dodson 1979
Agricultural	1-3 Butadiene (field)	Decreased nutrient loss	Pruen 1973
	Simazine lindane ceresan	Decreased soil respiration nitrification	Gaur and Misra, 1977
	Captan (nursery)	Decreased respiration nitrification	Agnihotri 1971
	Feedlot manure (laboratory incubation)	Increased Ca and Mg leaching	Amonzequar Fard *et al* 1980
Arctic	Methanol (laboratory and field)	Inhibited CO_2 efflux	Bryant and Parkinson, 1978

within the rooting zones There is good experimental evidence that both nutrient export and nutrient pools are useful ecosystem monitors

3.1 NUTRIENT EXPORT

The responses of nutrient export to both chemical and nonchemical stresses have been published Nutrient export can occur as particulates (resuspension, erosion), solutes (to groundwater or surface drainages), or gases (atmospheric losses) Depending upon the ecosystem and the stress applied, measurements useful in monitoring nutrient export differ For example, solute and gaseous nutrient losses may dominate uncultivated systems such as forests or prairie while particulates may be as important as solutes or gases in determining export from cultivated systems

Several studies showing effects of chemical and nonchemical stresses on terrestrial nutrient export rates are summarized in Tables I and II These effects have been reported in mesic, arid, and tundra ecosystems and in response to a variety of stresses. Increases in gaseous or solute nutrients are shown in each case

For example, there is a highly predictable relationship between annual output of dissolved nutrients and streamflow in a forested watershed. As Likens *et al* (1970) describe, clearcutting severely increased dissolved nutrient loss, net losses approximating eight times that of unaffected conditions. Nitrogen losses increased 160 fold and were attributed to lack of primary producer uptake and increased rates of decomposition (Smith *et al*, 1968) Similar results were obtained by trenching forest soil (Vitousek, 1977), NO_3-N loss increased significantly

Stress applied by girdling trees in a Liriodendron stand made it possible to decouple autotrophic-heterotrophic processes Response of the forest ecosystem increased NO_3-N and Ca loss beyond the rooting zone and increased CO_2 loss from soil (Edwards and Ross-Todd, 1979) Increased Ca and NO_3-N loss was also reported by Brown *et al* (1973) for the Oregon coast range as a result of cutting and slash-burning and by Aubertin and Patric (1974) in West Virginia as a result of clearcutting

Deposition of lead smelter baghouse dust upon Acer rubrum microcosms resulted in increased CO_2 efflux and dissolved nutrient (Ca, Mg, K, and NO_3-N) export from the intact excised soil (Jackson *et al*, 1978)

Increased CO_2 efflux resulted from hexachlorobenzene contamination of forest soil (Ausmus *et all*, 1979) and cadmium contamination of desert soil (Ausmus and Dodson, 1978) Arsenic contamination increased dissolved PO_4-P, NO_3-H, NH_4-N, and DOC loss from grassland soils (Jackson *et al.*, 1979) Feedlot manure additions to soil columns in the laboratory increased Ca and Mg leaching from the soil (Amoozequar-Fard *et al*, 1980)

Dose-response relationships have been established for several chemicals in laboratory-incubated, excised, intact microcosms (Draggon and Giddings, 1978; Ausmus *et al*, 1978, Jackson *et al*, 1977, 1979, Ausmus *et al*, 1979, Gilem *et al*, 1979). Complex effluents or process wastes have also been tested in this manner (Duke *et al*, 1977) and dose-response relationships established In general, nutrient export increases as a function of chemical contaminant concentration At higher doses, export rates may be

TABLE II

Nutrient pool changes as a function of chemical contamination

Ecosystem	Chemical (type study)	Nutrient pool changes	References
Agricultural	Tomato processing wastes (field)	Increased extractable PO4–P over fertilized and unfertilized soil	Timm *et al*, 1980
	Feedlot waste (field)	Increased NO_3–N extractable in soil though total N balance unaffected	Smith *et al*, 1980
	Pesticides (field)	Increased total litter pool	Ward and Wilson 1973
	Thiram (field)	Increased soil pools of Zn, Na, K, increased NH_4–H, decreased NO_3–N	Wainwright and Pugh, 1974
	Benoxyl (laboratory)	Increased NH_4–N, decreased NO_4–N in soil water	Foster and McQueen, 1977
	Benoxyl, dyrene, maneb	Organic N inorganic N increased	Mazur and Hughes, 1975
	SO_2	N S decreased in wheatgrass	Heitschmidt *et al*, 1978
		Decreased plant crude protein	Schwartz *et al*, 1978
	Acid Rain	Decreased microbial nutrient pools	Beath *et al.*, 1979
Forest	Heavy metals (field)	Depleted nitrogen in litter	Tyler, 1974
	Lead copper zinc (field)	Increased total litter pool	Anderson *et al* 1980
	Heavy metals (field)	Increased total litter pool	Ruhling and Tyler, 1973
	Nickel, copper	Increased litter pool	Freedman and Hutchinson, 1980a
	SO_2	Increased calcium, magnesium silicon potassium in leaf tissues	Ziegler, 1975
	Acid rain (field, laboratory)	C/N in soil increases	Tamm, 1976
	Heavy metals	Increased total litter pools, decreased soil productivity	Jackson and Watson, 1977
Grassland	Insecticide	Increased litter pool	Barrett, 1968

drastically due to the depleted nutrient pool in soil or due to inhibition of whole sets of processes such as litter decomposition (Jackson and Watson, 1977, Ausmus and O'Neill, 1979)

The sensitivity of nutrient cycling measurements (both export and pool sizes) relative to populations and community measurements was demonstrated by O'Neill *et al.* (1977) In both laboratory and field studies, nutrient cycling effects occured at lower chemical concentrations than measurable population or community effects Borman and Likans' (1979) analysis of the Hubbard Brook clearcutting experiment confirms these observations (Likens *et al*, 1967) From an analysis of data resulting from pilot studies of coal conversion processes sponsored by EPA-Industrial Environmental Research Laboratory (IERL), relative sensitivity of several biological and ecological measurements to complex chemical mixtures can be compared. Where comparable data are available, export of soil nutrients and CO_2 efflux from soil were more sensitive to chemicals than biological evaluations such as the Ames assay (Ames *et al*, 1975) or acute ecological effects tests

An additional argument for the importance of measuring ecosystem response to stress can be made from reclamation studies Bormann and Likens (1979), Parkinson *et al* (1980), and Walker and Adams (1958) have documented the recovery of nutrient cycling processes as biological reorganization begins McGill (1980) notes that N P ratios decrease as organic phosphorus increases whereas C and N accumulation within the soil is dependent upon carbon input rate and we would argue also dependent upon carbon form Edwards and Ross-Todd (1979) report that after tree girdling, total CO_2 efflux from the forest floor was not different from that in control plots while CO_2 respiration from soil (less roots and litter) was much greater Carbon loss from disrupted soil biochemical processes was evidently compensated for by root and litter carbon conservation

3 2 Nutrient pools

Nutrients are retained in terrestrial ecosystems in strata such as litter and soil and in biota both aboveground and belowground The concentrations of nutrients in biota, litter, and soil are affected by chemical contamination. Nutrient pool measurements may be increased or decreased by chemical contamination depending on the loading rate and the length of time that the inputs have occured For example, a stress causing litter decomposition disruption might be initially detected as increased litter nutrient pools for a period of time and subsequently as depleted pools (e.g., higher C/P) after nutrients are leached by physiochemical processes Alternatively, plant nutrient pools and production could be temporarily increased due to increased soil solution concentrations of nutrients caused by physical or chemical stress (Gersper and Challinor, 1975).

Representative studies showing the effects of chemical contamination on nutrient pools are listed in Table II. Several studies show changes in extractable or available soil nutrients under chemical contamination (Timm *et al*, 1980; Smith *et al.*, 1980; Wainwright and Pugh, 1974, Foster and McQueen, 1977; Mazur and Hughes, 1975). Tamm (1976) notes soil C/N increased in response to acid precipitation. Many studies ability of an ecosystem to persist might also be detected in the nutrient *pools of* ecosystem components (e g , litter or aboveground plant tissues) before structural components are permanently altered Figure 3 is a conceptualization of the soil nutrient pool and its nutrient export monitoring points The sensitivity of the potential exports (especially soil solution export even though it is very small compared to the total nutrient pool) will eventually reduce these pools in one or more ecosystem components Some sensitive measures of nutrient pool changes such as lignin nitrogen (Cromack *et al* , 1975) or C P (McGill, 1980) in litter or other substrates may allow periodic evaluation of ecosystem status. Cromack *et al* (1975) have shown that decomposition rates can be accurately predicted using recalcitrant carbon concentrations such as lignin Changing carbon and nutrient concentrations in tissues may be indicative of stress.

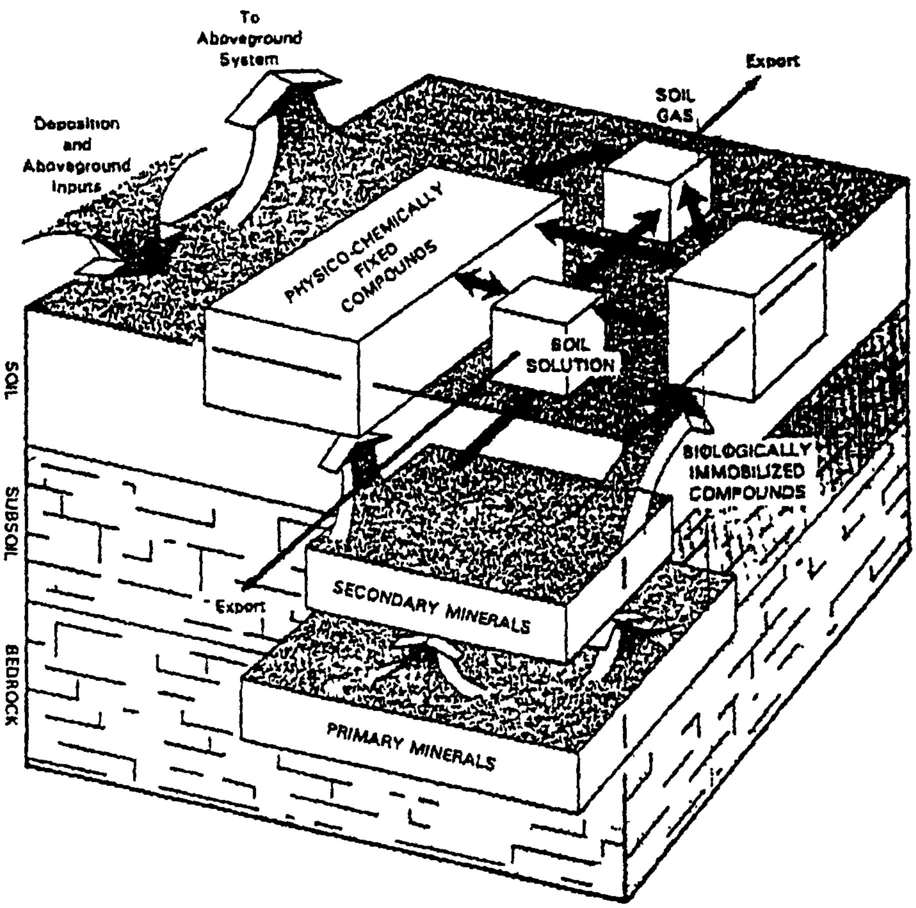

Fig. 3 Diagram of ecosystem nutrient flows.

show increased litter nutrient pools due to chemical disruption of cycling processes (Anderson *et al*, 1980), Ruhling and Tyler, 1973, Ward and Wilson, 1973, Barrett, 1968, Freedman and Hutchinson, 1980a, Hutchinson and Havas, 1978, Hutchinson and Whitby, 1974) Longer term (higher contamination levels) effects were shown by Tyler (1974) and Jackson and Watson (1977) who observed depleted nitrogen pools in litter due to heavy metals

Sulfur dioxide has been shown to at least temporarily increase Ca, Mg, Si, and K in leaf tissues (Ziegler, 1975) and to decrease crude protein (Schwartz *et al.*, 1978) Wheatgrass N S ratio was decreased by increased sulfur incorporated into tissues (Heitschmidt *et al*, 1978)

There is evidence that ratios of nutrients may be more sensitive or meaningful indices of chemical stress than nutrient pools alone For example, lignin/N of leaf tissue has been shown to be a more accurate predictor of leaf decomposition than lignin or C.N (Cromack *et al*, 1975) Ratios of nutrients derived from different reservoirs (e g, C.P) (carbon from atmospheric CO_2, phosphorus from minerals) may be very sensitive measures of ecosystem stress (McGill, 1980) Resolution of the best measures (e g, ortho-phosphate or organic phosphate, total carbon or recalcitrant carbon) is not yet possible from the data available

4. Conceptualization of Nutrient Cycling Disruption

The evidence presented above clearly demonstrates nutrient cycles are sensitive to chemical contamination An hypothesis of the mechanisms and consequences of nutrient cycling disruption is presented in Figure 4 Tamm (1976) has specifically applied a more specific hypothesis to the effects of acid rain on terrestrial ecosystems Our hypothesis states that litter-soil biochemical and microbial interactions are disrupted as a function of anthropogenic depositions As a result of these disruptions, temporarily greater solute and gaseous pools of nutrients are released These would potentially be available for plant uptake and/or export from the system depending upon the season, root activity and meteorological conditions (see Figure 4, Step 3) Decrease in biochemical and microbial interactions in litter-soil increases C/N, C/P, and lignin/nutrients providing for the inhibition of nutrient cycling processes Increased export of nutrients eventually lowers nutrient availability to producers, decreases nutrient retention within the system, and leads to long-term fertility and productivity loss at the site.

A temporary increase in primary production as a result of increased nutrient uptake may result in increased detrital inputs (see Figure 4, Steps 1 and 2), temporarily offsetting for a finite time period in the litter-soil nutrient depletion above However, these organic materials, if nutrient processes remain disrupted, will not be available for recycling to producers in appreciable concentrations or at the specific points in time when maximum uptake might occur in a natural system (Ausmus *et al*, 1977, Tyler, 1972)

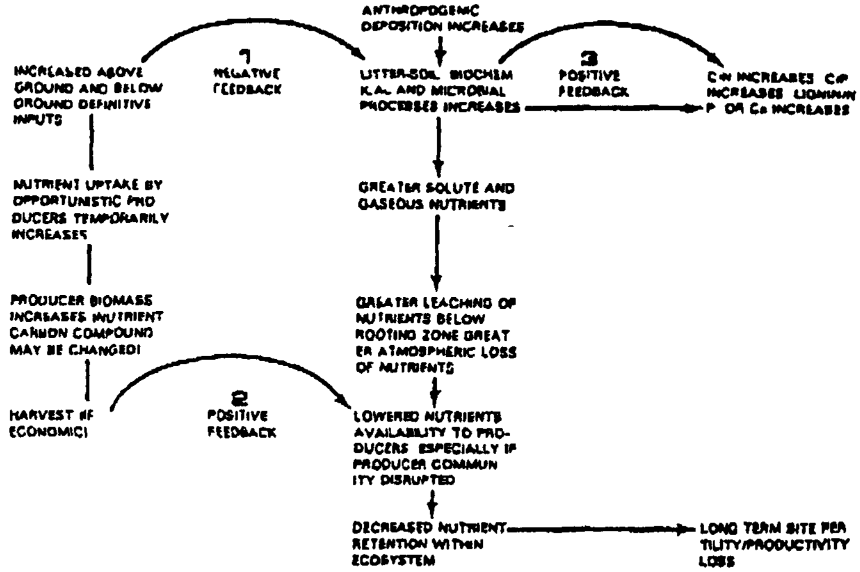

Fig 4 Hypothetical degradation of nutrient cycling processes within a terrestrial ecosystem.

5. Implementation of an Ecosystem Monitoring Program

Evidence is now available from various laboratories experimenting with a host of chemical contaminants which unequivocally substantiated the concepts of terrestrial nutrient cycling processes as sensitive indicators of polutant effects on ecosystems. Moreover, the level of evidence now available strongly suggests causative mechanisms of nutrient cycling response to chemical stress

Perhaps the delicate interrelationships taking place belowground among soil physico-chemical constituents, microorganisms (regulated by invertebrates), and plant roots are the most sensitive to chemical stress of the entire terrestrial ecosystem It is here, too, that many chemical contaminants accumulate because of the high adsorptive capacity of soils Thus, soil-microorganism-root interactions should indicate the initial and continual response of a terrestrial ecosystem under chemical stress Implementation of a monitoring technique using nutrient cycling processes will allow design criteria for monitoring programs to be satisfied The minimal costs of measuring nutrient export from ecosystem components would allow long-term implementation as a monitoring strategy Data from nutrient cycling processes measured in stressed ecosystems would allow the formulation of meaningful mechanistic hypotheses of nutrient disruption. These data could also be used to project future population and community effects (O'Neill *et al*, 1977) Such a monitoring strategy would allow rapid detection of ecosystem response to stress in time to apply appropriate control technologies or remedial actions and is applicable across terrestrial ecosystems and among a variety of chemical contaminants

Development of an ecosystem monitor would allow verification of physicochemical and microbial effects tests used in the laboratory under TSCA and FIFRA guidelines The parameters measured will provide a continuing documentation of ecosystem response to and/or recovery from stress which further allows determination of effectiveness of controls or remedial actions Finally, an ecosystem monitor provides feedback to both theory and understanding of ecosystems and to regulatory bodies for use in standard evaluation and guideline development

Such ecosystem monitors respond to the collage of contaminant inputs but do not (1) measure contaminant concentrations, (2) estimate bioaccumulation rates, or (3) necessarily reflect specific population or community changes over the short-term The implementation of ecosystem monitoring methods would allow cost-affective, long-term sensitive detection of ecosystem responses to and recovery from stress If effects were detected, other, more specific techniques could be employed to quantify the nature of the effects on communities or populations

6. Uniqueness of an Ecosystem Monitor

The concept of using an ecosystem monitor for determining effects on terrestrial ecosystems has been substantiated through the use of dose-response relationships which have been developed from field and microcosm studies For example, the Crooked Creek Watershed study (Jackson and Watson, 1977) was conducted in an area having a clearly defined concentration gradient of heavy metals Using unpolluted control sites for statistical comparison, changes in nutrient pools in soil, litter, and primary producers were strongly correlated with metal concentration This diruption of nutrient cycling processes was later confirmed using forest microcosms treated with heavy metal emissions (Jackson *et al*, 1977)

The Crooked Creek Watershed study where a polluant gradient was clearly established validates the concept of the ecosystem monitor presented above because

- Contaminant monitoring provided documentation of heavy metal loading rates and defined the pollutant gradient in the affected area
- Biological monitoring documented the effects of heavy metals on populations and communities (e g, invertebrates) within the affected area In addition, bioaccumulation in various species were established (e g, rodents)
- Ecosystem monitoring detected the disruptive effects of heavy metals on the entire ecosystem by measuring nutrient cycling processes This monitoring tool can also be used to determine whether the ecosystem is degrading further or compensating for the chemical-induced stress

Applying appropriate monitoring strategies to point-source pollution problems as described above may not be as difficult as their application to non-point source problems The current acid precipitation problem is a good case in point Without adequate historical information, contaminant monitoring (i e, acidity) and biological monitoring may not be useful for effects documentation This conclusion is based on the observation that acidity does not bioaccumulate, there is no clearly defined acidity gradient, and effects on species and communities would be extremely difficult to ascribe to acidity

An ecosystem level monitor should pose fewer interpretive limitations than either contaminant or biological monitors for non-point source pollution. Monitoring ecosystem processes such as nutrient cycling provides the ability to measure total response to stress In addition, synergistic effects (e.g, heavy metal-acidity) can be determined even though the internal mechanisms may not be obvious

7. Conclusions and Recommendations

A thorough review of chemical effects monitoring of terrestrial ecosystems indicates that ecosystem-level monitoring programs should be used in preference to biological monitoring Ecosystem-level processes can be measured and used in monitoring effects of pollutants on the total terrestrial ecosystem Potential monitors include (1) total ecosystem productivity, (2) system carbon metabolism, and (3) nutrient cycling processes and system retention efficiency.

Monitoring nutrient cycling processes have been experimentally verified to be sensitive indicators of ecosystem stress as a result of chemical or physical disruption. In both laboratory and field studies, nutrient cycling effects occured at lower chemical contaminant concentrations than measurable population or community effects. An ecosystem-level monitor may pose fewer interpretive limitations than either contaminant or biological monitors for non-point source pollution.

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WOB Comments on Draft Phase I RFI/RI Work Plan Operable Unit No. 10:

Page 1-4, Section 1.1, Volume I

What is the difference between Phase I and Phase 1? It states that Phase 1 is complete and Phase 2 is in progress, but the document is a Phase I document.

Page 2-5, Section 2.1.1.1, Volume I

D004-D011 wastes are now hazardous wastes due to toxicity under the Toxicity Characteristic (TC), and the Toxicity Characteristic Leaching Procedure (TCLP) is the test that is used to determine the toxicity characteristic. EP Toxicity is no longer in use.

In the last sentence on the page the word "solutions" is spelled incorrectly.

Page 2-7, Section 2.1.1.4, Volume I

The possible contaminants are listed as being plutonium, americium, and uranium. The hazardous wastes that are listed on page 2-5 (D001-D011, F001-F003, F009) should also be listed in this section as possible contaminants.

Page 2-97, Section 2.1.10.1, Volume I

The last sentence on the page should read "Building 460 dumpsters" instead of "these 460 dumpsters."

Page 2-116, Section 2.1.12.4, Volume I

In the last sentence on the page, the phrase "for this area" is repeated and one set should be deleted.

Page 2-120, Section 2.1.14.1, Volume I

It states that "IHSS 210 is currently being used in a 90-day storage unit", but this contradicts the next sentence which states "as of May 31, 1988, all hazardous waste was removed from IHSS 210." If the unit is being used as a 90-Day storage unit, hazardous wastes are still present ("in" should be replaced by "as").

Page 2-125, Section 2.1.15.1, Volume I

In the first sentence, the word "results" is spelled incorrectly.

The statement that "saltcrete results from evaporation of liquid process water" is not completely correct. Saltcrete is made from mixing salt and brine resulting from the evaporation of liquid process waste water with portland cement.

The statement that "materials will be removed from the 904 Pad by October 1991 is no longer correct.

In the last paragraph, the statement that "incomplete solidification of the waste material which results in failure of the container" is incorrect. Weathering destroyed the original container, which resulted in the release of the incompletely solidified waste.

Page 2-126, Section 2.1.15.1, Volume I

The word "cubic" is spelled incorrectly.

Page 2-126, Section 2.1.15.2, Volume I

In the last paragraph, the second sentence should read "The runoff may result in chronic low levels of contaminates being released into Pond B-5 and this discharge from the pond could violate the NPDES permit."

Page 2-128, Section 2.1.15.4, Volume I

The word "uranium" is spelled incorrectly where it states "uranium 238."

Page 2-134, Section 2 1 16 1, Volume I

The statement that "saltcrete , a material similar in nature to pondcrete resulting from evaporation of liquid process waste" is not completely correct Saltcrete is made from mixing salt and brine resulting from the evaporation of liquid process waste water with portland cement.

Page 2-140, Section 2 1 16 1, Volume I

The statement that "there is a high probability that leakage of material will continue until all materials are removed" is incorrect. All materials that had poor containment have either been transferred to metal boxes or have been rebagged in thick plastic bags, and there have been few spills since the rebagging

Page 5-7, Section 5 7 1, Volume I

It states that "although RCRA regulations will direct the RI at OU10, CERCLA will also be considered for guidance because is specifies in greatest detail the steps that should be followed for selection of remedial alternatives" ("it" should replace "is"). RCRA allows more flexibility than CERCLA, and all steps under CERCLA should not necessarily be followed if they are not needed.